Int J Life Cycle Assess (2014) 19:52–68
DOI 10.1007/s11367-013-0638-5

Comparing direct land use impacts on biodiversity of conventional and organic milk—based on a Swedish case study

Carina Mueller · Laura de Baan · Thomas Koellner

Received: 1 February 2013 / Accepted: 29 July 2013 / Published online: 15 August 2013

Abstract
Purpose Halting the loss of biodiversity while providing food security for a growing and prospering world population is a challenge. One possible solution to this dilemma is organic agriculture, which is expected to enhance biodiversity on the farmland. However, organic products often require larger areas. This study demonstrates how we can quantify and compare the direct land use impacts on biodiversity of organic and conventional food products such as milk.
Material and methods This study assessed direct land use impacts of 1 l of milk leaving the farm gate. Inventory data on land occupation were extracted from a life cycle assessment study of 15 farms in southern Sweden. Direct land use change data were derived from the FAO statistical database. Spatially differentiated characterization factors of occupation (CF_{Occ}) and transformation (CF_{Trans}) were calculated based on the relative difference of plant species richness on agricultural land compared to a (semi) natural regional reference. Data on plant species richness and regeneration times of ecosystems (for calculating transformation impacts) were derived from a literature review. To account for differences in biodiversity value between regions, a weighting system based on absolute species richness, vulnerability and irreplaceability was applied.
Results and discussion Organic milk had a lower direct land use impact than conventional milk, although it required about double the area. Occupation impacts dominated the results and were much smaller for organic than conventional milk, as CF_{Occ} of organic land uses were considerably smaller. For transformation impacts, differences between the two farming practices were even more pronounced. The highest impacts were caused by soymeal in concentrate feeds (conventional milk) due to large-scale deforestation in its country of cultivation (i.e. Brazil and Argentina). However, lack of reliable data posed a challenge in the assessment of transformation impacts. Overall, results were highly sensitive to differences in land occupation area between farms, the CF_{Occ} and assumptions concerning transformed area. Sensitivity and robustness of results were tested and are discussed.
Conclusions Although organic milk required about twice as much land as conventional, it still had lower direct land use impacts on biodiversity. This highlights the importance of assessing land use impacts not only based on area but also considering the actual impacts on biodiversity. The presented approach allows to quantify and compare hot- and coldspots in the agricultural stage of milk production and could potentially also be applied to other agricultural products. However, more research is needed to allow quantification of indirect land use impacts.

Keywords Biodiversity · Land use · Life cycle impact assessment · Milk · Organic agriculture

Responsible editor: Ralph K. Rosenbaum

Electronic supplementary material The online version of this article (doi:10.1007/s11367-013-0638-5) contains supplementary material, which is available to authorized users.

C. Mueller (✉) · T. Koellner
Faculty of Biology, Chemistry and Geosciences, University of Bayreuth, GEO II, Room 1.17, Universitaetsstr. 30, 95440 Bayreuth, Germany
e-mail: Carina.M.Mueller@web.de

L. de Baan
Institute of Environmental Decisions, Natural and Social Science Interface, ETH Zurich, Universitaetsstr. 22, 8092 Zurich, Switzerland

1 Introduction

One of the most important drivers of global biodiversity loss is land use change or habitat destruction (Sala et al. 2000; Foley et al. 2005; Pereira et al. 2010), which is indirectly linked to increased food and energy demand by a growing and prospering world population (Tilman 2001). Providing food security

for the expected future human population of 9 billion people (Zhang 2008) while halting the loss of biodiversity is a challenge. Over the past years, there has been a growing debate about whether land sparing (i.e. agricultural intensification to minimize pressure to convert wild nature) or wildlife-friendly farming (i.e. increasing biodiversity on farmland which might reduce agricultural yields) is the more promising solution (Tscharntke et al. 2012; Chappell and LaValle 2011; Ewers et al. 2009). Organic farming practices are found to increase species abundance (Crowder et al. 2010; Rahmann 2011) and species richness of many taxa (Bengtsson et al. 2005; Fuller et al. 2005; Gomiero et al. 2011) as well as the presence of endangered species (van Elsen 2000) compared to conventional farming. This positive effect on biodiversity points to the management principles of organic agriculture, which prohibit the use of chemical fertilizer or inorganic pesticides (Hole et al. 2005) and are characterized by a lower stocking rate (Hansen et al. 2001) and a more varied crop rotation (Maeder et al. 2002).

Especially in Europe, efforts to preserve the high biodiversity value of long-established agricultural ecosystems are high (Green 2005). Therefore, subsidies for organic farming were introduced in the early 1990s (Stolze and Lampkin 2009). In 2005, the European Union's (EU plus national funds) support of organic farming increased to 17 % of the yearly expenditures (or a total of 0.6 billion Euro) dedicated to agri-environment measures (European Union 2007). In order to assess the usefulness of these large amounts of subsidies for biodiversity conservation, the overall impacts of organic and conventional products on biodiversity need to be quantified.

An important decision support tool in this context is life cycle assessment (LCA), which allows the quantification of a range of environmental impacts throughout all stages of a product's (often globally distributed) life cycle (e.g. Milà i Canals et al. 2007). Several approaches towards the assessment of direct land use impacts on biodiversity have been published. However, these studies are either only applicable to one or two world regions, and thus cannot capture land use impacts occurring in many world regions (e.g. Koellner 2000; Koellner and Scholz 2007; Michelsen 2008; Schmidt 2008a), or do not capture differences among different farming systems (e.g. Weidema and Lindeijer 2001; de Baan et al. 2013; Mattila et al. 2012).

In this study, we quantify and compare the direct land use impacts on biodiversity of organic and conventional milk. We therefore further develop the methodology presented in de Baan et al. (2013) and apply it to a case study of milk production in Sweden, where organic milk was shown to require considerably larger areas compared to conventional milk (Cederberg and Flysjoe 2004). Milk is an interesting example because its land use is distributed over many world regions via the supply chain of concentrate feed (Cederberg and Mattson 2000). Thus, a regionalized global approach (as suggested in de Baan et al. 2013) is needed to accurately assess biodiversity impacts, due to its heterogenous distribution and variable response to land use. In addition, the composition and origin of feedstocks for organic and conventional milk vary substantially. For producing organic milk in the EU, more than 60 % of the daily intake of cows has to be roughage such as silage, hay or pasture and more than 50 % of the feed has to be produced on the farm itself (EC 2007). Conventional dairy cows are fed more on concentrate feed compared to organic cows, and the feedstock are to a larger degree imported from sub-/tropical countries (Cederberg and Mattson 2000). In these countries, large areas of natural vegetation of high species richness and endemism (Brooks et al. 2006; Kier et al. 2005; Myers et al. 2000) have been transformed in the last decades (FAOSTAT 2012a; Mayaux et al. 2005).

Milk is not only an interesting case to illustrate the trade-offs between area requirement and impacts on biodiversity but also an important commodity. The consumption of milk products is projected to rise considerably in the next decades (Delgado 2003). Many LCA studies have been conducted on the comparison on organic and conventional milk. In terms of energy use, eutrophication and global warming potential no clear result of organic compared to conventional milk can be found (Thomassen et al. 2008; Flysjoe et al. 2012). In contrast, land use (in terms of area) was found to be considerably larger for organic milk than for conventional milk (Cederberg and Mattson 2000; Thomassen et al. 2008; van der Werf et al. 2009). However, the impacts of land use on biodiversity have not yet been incorporated into LCA studies of milk (Yan et al. 2011).

2 Material and methods

2.1 Description of compared agricultural systems

Data on land occupation area and country of cultivation of fodder crops were taken from a study on dairy farms in south-west Sweden (Cederberg and Flysjoe 2004), following the same approach as Cederberg and Mattson (2000). In this study, we considered nine high-intensity conventional farms and six organic farms. In terms of total farm area, the two agricultural systems were comparable; however, the conventional farms had to purchase more concentrated feed as they had more cows on their farms (Table 1). The higher share of roughage feed on the diets of organic cows results in lower milk yields of organic compared to conventional cows. The functional unit (FU) of the study was "1 kg of energy corrected milk" leaving the farm gate, i.e. transportation and processing of raw milk were excluded (Cederberg and Flysjoe 2004). Partitioning of environmental impacts (i.e. allocation) between meat and milk as well as co-products in purchased concentrate feed, such as meal and oil from rapeseed, was

based on their economic value (Cederberg and Flysjoe 2004). Additionally, direct land use impacts were ascribed to the region in which the crop was most likely cultivated according to Cederberg and Flysjoe (2004) and for organic soy according to FiBL (2012). In this study, it was assumed that the land was occupied for one whole year for most crops, as in temperate latitudes only one fodder crop can be grown per year and oil palm fruit, meadows and pastures are cultivated permanently (Milà i Canals et al. 2013). Only after the harvest of soy another fodder crop can be grown in the same year. Thus, about 25 % double-cropping of soy was assumed (suggested in Dalgaard et al. 2008) which was in contrast to Cederberg and Flysjoe (2004) who did not account for any double-cropping.

2.2 Land use assessment framework

Following the framework of the United Nations Environment Programme/Society of Environmental Toxicology and Chemistry Life Cycle Initiative (Milà i Canals et al. 2013; Koellner et al. 2013a, b), we distinguished two land use impacts: land occupation (using land) and land transformation (changing the land use) in this study (Milà i Canals et al. 2013; Koellner et al. 2013a, b). Permanent impacts (i.e. irreversible damages to ecosystems) were not considered here. Characterization factors of land occupation, CF_{Occ}, were calculated as the difference in relative plant species richness (S_{rel}) of a reference system and a land use type (LU) *i* per biome *j* (de Baan et al. 2013; Koellner et al. 2013a, b). As globally available biodiversity data focus on the species level and species numbers (Curran et al. 2011), we chose *relative* species richness of a certain land use type *i* compared to a reference as biodiversity indicator. S_{rel} of the reference system (ref) was set to 100 %=1 (see Section 2.4.1).

Table 1 General characteristics of the two farm types

	Organic	Conventional
Number of studied farms	6	9
Arable land (ha farm^{-1})	63	70
Natural meadows (ha farm^{-1})	19	11
Number of cows per farm	39	65
Stocking rate (LU[a] ha^{-1} year^{-1})	0.9	1.2
Total purchased feed (kg cow^{-1} year^{-1})	1,457	2,951
Milk yield (kg ECM[b]/cow×year)	9,400	10,100

[a] Lifestock unit: corresponds to one dairy cow including a calf younger than 1 month, or six calves aged between 1 and 6 months, or three heifers older than half a year

[b] Energy corrected milk

$$CF_{Occ,LU\,\mathrm{i,j}} = S_{rel,Ref\,\mathrm{j}} - S_{rel,LU\,\mathrm{i,j}} = 1 - S_{rel,LU\,\mathrm{i,j}} \tag{1}$$

Natural vegetation was chosen as reference. In many European countries, only few anthropogenic undisturbed habitats remain, so they cannot be considered natural habitats (SOER Synthesis 2010). Consequently in the following, we will refer to the reference situations as "(semi)-natural", as proposed in de Baan et al. (2013).

The numeric values of CF_{Occ} normally take on values between zero and one (expressing detrimental impacts), but can sometimes also be negative (expressing beneficial impacts, de Baan et al. 2013). The occupation impact can then be calculated as the product of CF_{Occ}, the cultivated area A_{Occ} and the duration of the occupation process t_{Occ} (de Baan et al. 2013; Koellner et al. 2013a, b).

$$Occupation\ impact_{LU} = A_{Occ} \times t_{Occ} \times CF_{Occ} \tag{2}$$

Transformation impacts are assessed accordingly with the transformed area A_{Trans} (see Section 2.2) and the time a system requires to regain the ecosystem quality of the reference (t_{reg}) after an anthropogenic disturbance (de Baan et al. 2013; Koellner et al. 2013a, b).

$$\begin{aligned} Transformation\ impact_{Ref \to LU} &= A_{Trans} \times CF_{Trans} \\ &= A_{Trans} \times \frac{1}{2} \times CF_{Occ} \times t_{reg} \end{aligned} \tag{3}$$

The Biodiversity Damage Potential (BDP) of land use can then be calculated as the sum of the transformation impact and the occupation impact over all land use types *i* and biomes *j* (Koellner and Scholz 2007).

$$\begin{aligned} BDP = &\sum Transformation\ Impact_{Ref \to LU_{i,j}} \\ &+ \sum Occupation\ Impact_{LU_{i,j}} \end{aligned} \tag{4}$$

2.3 Inventory analyses

In the study of Cederberg and Flysjoe (2004), transformation impacts are not considered. Thus, we calculated the inventory data for transformed area as proposed by Milà i Canals et al. (2013). This approach only associates direct land transformation with a fodder crop if (1) in its country of origin the harvested area of this specific crop increased in the last 20 years and if additionally (2) the area of its land use type (i.e. arable land, permanent crops or meadows and pastures) increased. In case these two conditions applied, the transformed area for every occupied hectare and year was calculated by dividing the increase in land use type area over

the last 20 years by the current area of this land use type (as proposed in Milà i Canals et al. 2013). The type of land that was transformed was proportionally assigned based on the decrease in area of all land use types. Historical data on country-specific developments of agricultural production areas were extracted from the statistical database of the Food and Agriculture Organisation (FAOSTAT 2012a, b For grass pellets, data were extracted from Statistics Denmark (StatBank Denmark 2012) as no information was given in the FAOSTAT database. For organic land use types, no specific information on land use change was available and standards of the International Federation of Organic Agriculture Movements only prohibit the conversion of areas with high conservation value 5 years before organic farming starts (IFOAM 2012). Therefore, we assumed organic farming to result in the same amount of transformed area per cultivated hectare as conventional farming. To account for yearly variation in land use, 5-year averages around the years 1987 and 2007 were calculated, respectively (as suggested in Milà i Canals et al. 2013). As data on forest area before 1990 were not available for all countries, pre-1990 areas were linearly extrapolated using data from the years 1990 to 2009. This seems a feasible estimate, as the forest conversion rate after 1990 followed a linear relationship for the countries considered in this study. Furthermore, within the timeframe in question (1985–2009) the same drivers were prevalent, as demonstrated in various studies (e.g. Fearnside 2005; Gasparri et al. 2008; McMorrow and Talip 2001).

Forest is the only natural terrestrial land cover type included in the FAOSTAT database (2012a). Most Brazilian soy is cultivated on land where naturally savannahs prevail (Fearnside 2001). Savannahs still fall under the definition of "forest" according to the FAOSTAT database, as they contain vegetation with trees exceeding 5 m and a canopy cover denser than 10 %. Consequently, these data were seen as appropriate for estimating conversion from natural vegetation within the biome sub-/tropical grass-/shrublands and savannahs.

2.4 Impact assessment

To account for spatial differences in ecological responses to land use, region-specific reference situations and regeneration times (see Eq. 3) were used and impacts were calculated separately for each biogeographic region (as suggested in de Baan et al. 2013). Biomes defined by Olson et al. (2001) and recommended to use for land use impacts in LCA on global scales by Koellner et al. (2013a, b) were used as biogeographic units. CORINE Plus (elaborated by Koellner and Scholz 2008) was used for land use classification as it allowed the discrimination of organic and conventional land use types (Koellner et al. 2013a, b).

2.4.1 Biodiversity indicator

The analysis of a biodiversity measure was restricted to vascular plants because data availability for organic land use types is relatively good for this taxon (Hole et al. 2005). Several data sources were combined here to quantify plant species richness of relevant land use types and reference situations: the Countryside Survey of the United Kingdom (Carey et al. 2007), the report "Biodiversity of Saxonian arable land" (Kreuter 2005), the biodiversity monitoring data of Switzerland (BDM 2004) and the databases used in Koellner (2000) and de Baan et al. (2013). As these data sources only contained little data on organic land uses or data from the biome sub-/tropical grass-/shrubland and savannahs, a literature search in the Web of Science database was performed. Overall, this search resulted in 66 studies (see Online Resource for a full bibliography) providing 111 data points for the different land use types and 53 data points for the reference situations in three different biomes of feedstock production for Swedish milk. As sampling area varied strongly among studies, sampled species richness (S) was standardized to an area (A) of 100 m^2 using the transformed power model of the species–area relationship proposed in Kier et al. (2005):

$$S_{100\mathrm{m}^2} = S_{Sampled} \times \left(\frac{A_{100\mathrm{m}^2}}{A_{Sampled}}\right)^z \tag{6}$$

where z is the species accumulation factor (see ESM, Table 1 for results of each data point). For each biome, z values were obtained from Kier et al. (2005).

2.4.2 Regeneration time

The time required for the regeneration after human interventions stop is used for the calculation of transformation impacts (Eq. 3). To derive regeneration times of different land use types, a literature search of peer-reviewed studies was conducted. Only investigations located below 1000 m a.s.l. analysing regeneration of plant species richness after human disturbances (e.g. logging or agricultural use) were included. However, for the regeneration times needed for this study only very few data points (one to five per land use type and biome) could be found (Table 2). Therefore, alternative regeneration times based on Dobben et al. (1998) were calculated as well for a comparison. These regeneration times have been used in previous LCA studies on land use (e.g. Milà i Canals et al. 2013), and are based on broad estimates of biomass regeneration dependent on altitude and latitude after clear-cutting. To make this estimate comparable to the empirical data, biome-specific regeneration times were calculated depending on each biome's area percentage (quantified with ArcGIS) in three different altitudinal and six different latitudinal zones. These values were considered to represent regeneration times of arable land ($t_{reg,arable,Dobben}$). Thus, the regeneration time

Table 2 Regeneration times by land use type

	Based on Dobben et al. (1998) t_{reg} (years)	Mean of empirical studies t_{reg} (years)	*n*	Source
Biome sub-/tropical moist broadleaf forests				
Pasture/meadow	45	33 [a]	5	(Aide et al. 1995, 1996; Guariguata et al. 1997; Letcher and Chazdon 2009)
Permanent crops	42	35[a]	3	(Grau et al. 1997; Pascarella et al. 2000; Zimmermann et al. 2007)
Biome temperate grass-/shrublands and savannahs				
Arable	–	45 [b]	1	(Scott and Morgan 2012)
Biome sub-/tropical grass-/shrublands and savannahs				
Arable	57	–		

n number of data points per biome and land use type

[a] All studies were conducted in Middle or South America

[b] From Australia

($t_{reg,LUi,Dobben}$) of every land use type *i* was then calculated using the ratio of the CF_{Occ} per biome *j* (see Section 2.1) as follows:

$$t_{reg,LU\mathrm{i,j},Dobben} = \frac{CF_{Occ,LU_{\mathrm{i,j}}}}{CF_{Occ,arable}} \times t_{reg,arable\mathrm{i,j},Dobben} \quad (7)$$

It was assumed that the regeneration rate, defined as the change of $CF_{Occ,\ LUi}$ with t_{reg}, of each land use was constant within each biome.

2.4.3 Statistical analysis

No studies were found which had investigated the influence of organic and conventional farming practice and the species richness on comparable (semi-)natural references together. Therefore, we performed a resampling procedure with 1,000 bootstrap samples (with replacement) to obtain several statistical parameters (e.g. median, quartiles, 95 % confidence intervals) of the relative species richness S_{rel}. For each land use type, bootstrapping combined all plant species richness data points randomly with (semi-) natural reference data points from the same biome.

Analyses of variance (ANOVA) was applied to test differences in means of S_{rel} among the factors land use type, biome and farming practice as well as their interactions. Prior to statistical analyses, bootstrapping results were log-transformed to improve fit to a normal distribution.

As data did not satisfy the assumptions of the ANOVA (i.e. normal distribution and homogenous residual variance), robustness of results was evaluated by additionally conducting the Kruskal–Wallis test to analyse the influence of the three factors (without interaction) on medians of S_{rel}. The Mann–Whitney *U* test revealed significant differences in median S_{rel} between pairs of land use types. All data were performed using R 2.13.0 (R Development Core Team 2011) and the R package "boot" (Angelo and Ripley 2012) for the bootstrapping.

2.4.4 Biodiversity weighting factor

Expressing species richness in relative instead of absolute terms allowed the comparison of land use impacts in various ecoregions with different data qualities (Kier et al. 2005; Koellner et al. 2013a, b). To account for differences in absolute species numbers and conservation value between ecoregions, a weighting system was applied, as proposed in Weidema and Lindeijer (2001) which was calculated separately for each ecoregion. Following global biodiversity prioritisation concepts, the biodiversity weighting factor (BWF) was based on *absolute species richness*, *irreplaceability* and *vulnerability* (see review by Brooks et al. 2006).

The three indices to quantify biodiversity value of each ecoregion were calculated as follows. Absolute species richness (*S*) was calculated as area-corrected total number of amphibian, reptile, mammal and bird species per ecoregion (derived from Kier et al. 2005). Irreplaceability was quantified as the area-corrected number of strict endemic species of amphibians, reptiles, mammals and birds (EndS*)*. For endemism, these are the only taxonomic groups where data per ecoregion are available. For consistency, the same selection of taxonomic groups was also chosen for species richness, and data on plants were excluded. Vulnerability was expressed as the 'Conservation Risk Index' (CRI), which is calculated as the ratio of converted ecoregion area in per cent to protected ecoregion area in per cent (developed by Hoekstra et al. 2005). The latter concept assumes that the more area is occupied the more damaging an occupation or transformation will be for the remaining ecosystem (Koellner 2000). To prevent division by zero, all values below 1 % were set to 1 % as such low habitat proportions are likely below the threshold necessary

for long-term species conservation (Swift and Hannon 2010). The same procedure was performed for converted area as the impact of converting 1 % of a habitat is assumed to be negligible in terms of species conservation.

Data to calculate all indices were extracted from the Terrestrial Ecoregions Base Global Dataset (Olson et al. 2001). As suggested by Weidema and Lindeijer (2001), these three measures of conservation prioritisation were linked by multiplication for each ecoregion *k*. To give each measure equal weight in the multiplication, they were all normalised (expressed with an asterix * in equation) to range between 1 and 10 (Table 3):

$$BWF_{\mathrm{k}} = \left(\frac{S_{\mathrm{k}} \times 9}{S_{max}} + 1\right) \times \left(\frac{EndS_{\mathrm{k}} \times 9}{EndS_{max}} + 1\right) \times \left(\frac{CRI_{\mathrm{k}} \times 9}{CRI_{max}} + 1\right) = S_{\mathrm{k}}{}^{*} \times EndS_{\mathrm{k}}{}^{*} \times CRI_{\mathrm{k}}{}^{*} \quad (8)$$

Overall, the weighted BDP was calculated as follows:

$$weighted\ BDP_{\mathrm{k}} = \left(\sum TI_{Ref \to LU_{\mathrm{i,k}}} + \sum OI_{LU_{\mathrm{i,k}}}\right) \times BWF_{\mathrm{k}} \quad (9)$$

3 Results

3.1 Characterization factors for biodiversity damage potential

For almost all land use types, medians of CF_{Occ} were significantly greater than zero, indicating detrimental impacts on biodiversity (see Figs. 1 and 2, and ESM Fig. 1, Table 2 and Table 6). Only the median of organic pasture/meadow was not significantly different from zero and showed a trend towards a negative value (i.e. beneficial impact). Overall, meadows/pastures were least harmful to biodiversity, followed by permanent crops. Arable land had the highest impact on biodiversity, which was most pronounced in the biome sub-/tropical moist broadleaf forests. A significant effect of land use type, biome and farming practice on CF_{Occ} was shown with ANOVA and Kruskal–Wallis test (results not shown). The ANOVA also revealed that the influence of land use type on CF_{Occ} was significantly affected by biome and farming practice.

In all studied land use types and biomes, organic farming practices were less detrimental for biodiversity than conventional farming practices. Medians of CF_{Occ} of conventional compared to organic pasture/meadow showed a 30 % higher potentially disappeared fraction (PDF) of species per square meter and year. In the median, 45 % less species were found on conventional arable land in the biome temperate broadleaf and mixed forests than on the organic equivalent. In the biome sub-/tropical grass-/shrublands and savannahs, this difference amounted to about 40 %.

Table 3 Results of normalisations of species richness (S), endemic species richness (EndS) and Conservation Risk Index (CRI) and their product, the BWF per ecoregion

Biome	Ecoregion	S[a]	EndS[a]	CRI[a]	BWF
Sub-/tropical moist broadleaf forest	Peninsular Malaysian rain forests	4.6	1.5	3.0	18.1
Temperate broadleaf & mixed forests	Atlantic mixed forests	2.4	1.0	7.8	18.6
	Baltic mixed forests	2.4	1.0	5.4	12.8
	Sarmatic mixed forests	2.1	1.0	1.6	3.4
Sub-/tropical grass-/shrublands and savannahs	Cerrado	3.3	1.8	5.4	31.7

Sources: Olson et al. (2001) and own calculations. A full list of BWF per ecoregion can be found in Online resource Table 6

[a] Results of normalisations

3.2 Impact of occupation

Although organic milk required about double the amount of agricultural land to produce 1 kg of milk (Fig. 3a), the occupation impact of organic milk was only half the one of conventional milk (Fig. 3b). CF_{Occ} of organic land use types were always considerably lower than the conventional ones thus leading to smaller occupation impacts. In addition, the different composition of the feedstock, with larger shares of roughage feed and grazing for the organic cows and larger shares of concentrate feed for the conventional cows, considerably influenced the result (ESM, Table 3).

Arable land and more precisely grains and legumes/temporary grasses in the biome temperate broadleaf and mixed forest largely dominated the impact for both farming practices (Fig. 4). For the fodder crop category legumes/temporary grasses 100 and 80 % of the impact was due to ley for conventional and organic milk, respectively. Both conventional and organic farms cultivated ley on about 60 % of their on-farm arable land (Cederberg and Flysjoe 2004). Grains accounted for more than 75 % in the organic and about 50 % in the conventional concentrate feed (ESM, Table 3). This explains the large land use and occupation impact allocated to this crop type. In contrast, rapeseed, sugar beet and palm oil have relatively small shares in the occupation impact of conventional milk, although they constitute about 37 % of the concentrate feed. Due to their form as co-products of the sugar/oil/starch production, only a small part of the environmental burden was allocated to them, minimizing their impact. Although organic milk required a relatively large area of

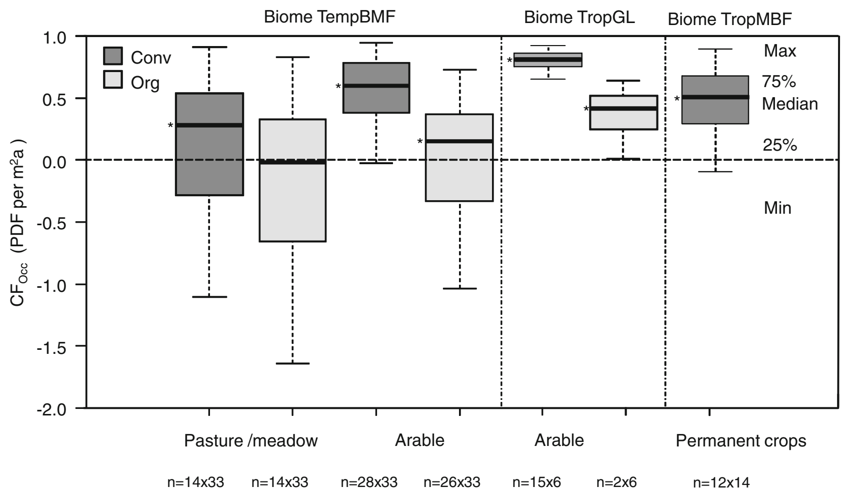

Fig. 1 Box and whisker plot of bootstrapped characterization factors of occupation (CF_{Occ}) for each land use type, farming practice and biome. *TempBMF* temperate broadleaf and mixed forests; *TropGL* sub-/tropical grass-/shrublands and savannahs; *TropMBF* sub-/tropical moist broadleaf forests. *Asterisks* mark significant differences in medians compared to the reference which is defined as $CF_{Occ}=0$ (Mann–Whitney U test, $p<0.05$). n is the total number of data points used for the bootstrap of CF_{Occ}; the first number stands for the number of species richness data points per land use type and the second for the data points per reference. $n=14\times33$, e.g. means that 14 data points on the species richness of the land use type were combined with 33 data points for the reference situation, which resulted in a total of 462 data points for CF_{Occ}

pastures and meadows, the occupation impact was slightly negative (i.e. beneficial) due to their negative median CF_{Occ} value. In the organic feedstock, some conventionally produced grains are included. This is the reason why grains and legumes have a similar occupation impact (Fig. 3b), although larger areas are required for legume production than for grain production (Fig. 3a).

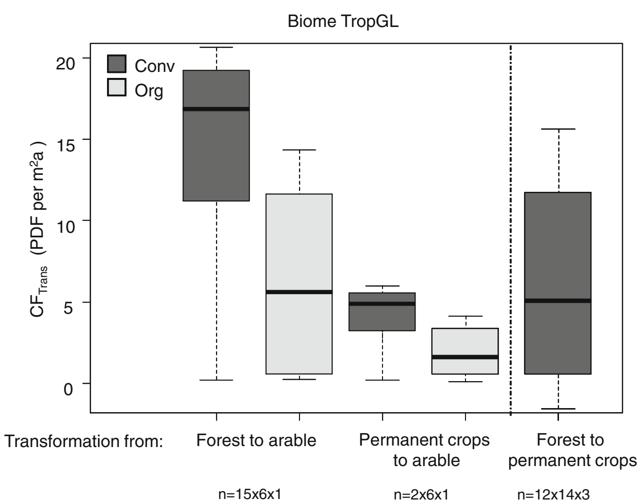

Fig. 2 Box and whisker plot of characterization factors of transformation (CF_{Trans}) for each farming practice, biome and land use type regenerating to after human abandonment. *TropGL* sub-/tropical grass-/shrublands and savannahs; *TropMBF* sub-/tropical moist broadleaf forests. Shown is only the variation due to differences in species richness between data points. See Online Resource Fig. 1 for variation in species richness per land use type and biome

3.3 Impact of transformation

Conventional milk had a more than threefold larger transformation impact than the same amount of organic milk, regardless of the method used for estimating regeneration times (Fig. 5). In this study, soy and palm oil were the only fodder crops originating from countries, where in the last 20 years the harvested crop area as well as the area of the corresponding land use type (i.e. arable land, pastures/meadows or permanent crops) increased. Soy was responsible for the majority of the transformation impact (99 % for conventional milk and 98 % for organic milk). In contrast, palm oil caused only a very small impact because palm kernel expels have only little economic value and the quantities in the concentrate feeds were small (3 % for conventional milk and 4 % for organic milk in concentrates). The land use change from natural vegetation to each harvested hectare of crop was the same for the conventional soy from Brazil (0.20 ha converted/harvested ha per year) and palm oil from Malaysia (0.20 ha converted/harvested ha per year), but slightly smaller for the organic soy from Argentina (0.14 ha converted/harvested ha per year). Argentina was the only country in the milk value chain, where one agricultural land use type, permanent crops, was converted into another agricultural land use type (ESM, Table 4). Thus the transformation impact of Argentinian soy is lower than Brazilian soy as the regeneration time of arable land to permanent crops is smaller than from arable land to secondary vegetation. Also, the much higher yields of conventional compared to organic soy were counterbalanced by the lower CF_{Trans} of organic arable land in this biome. As a consequence, the difference in the transformation impact

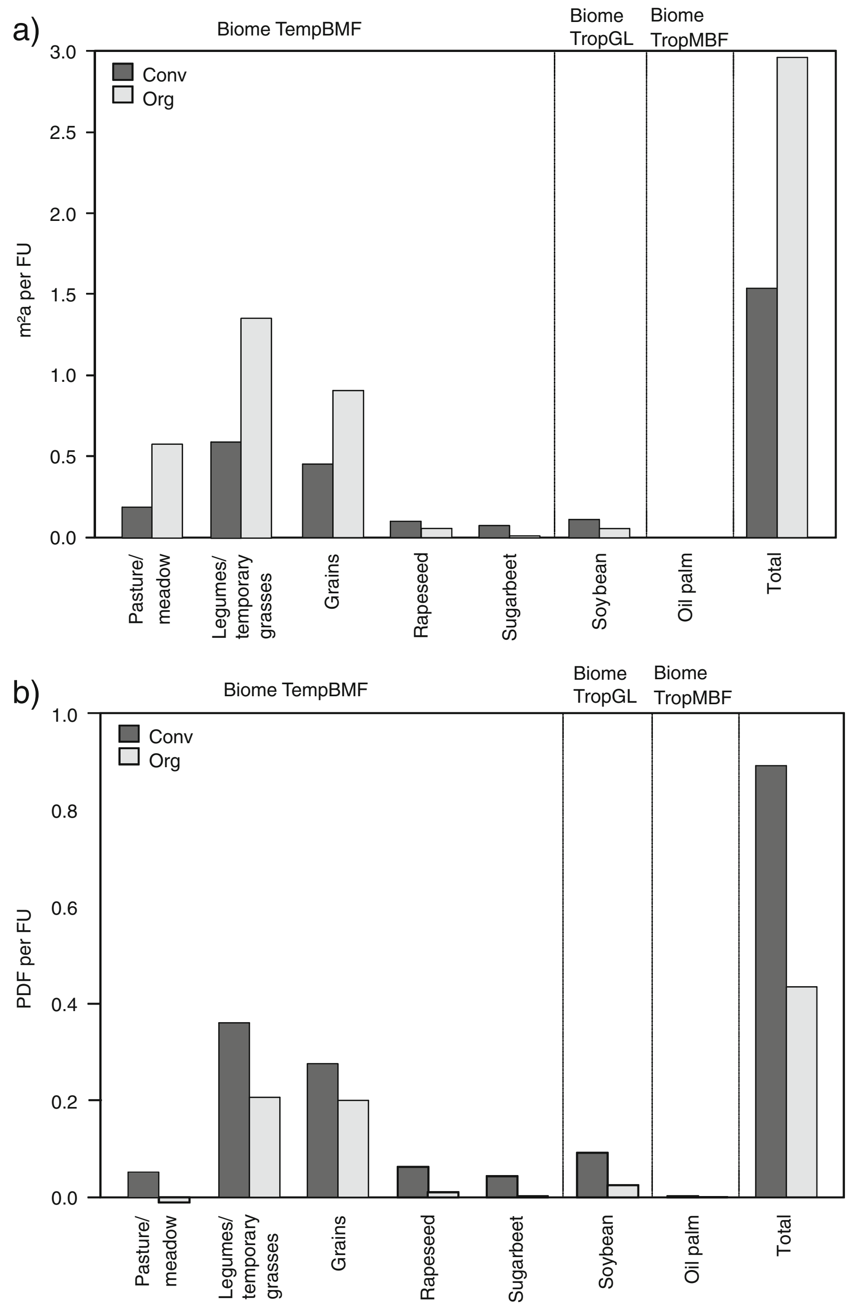

Fig. 3 Contribution to **a** *land occupation* and **b** *occupation impact* of different fodder crop types per biome TempBMF temperate broadleaf and mixed forests, TropGL sub-/tropical grass-/shrublands and savannahs and TropMBF sub-/tropical moist broadleaf forests, calculated with non-weighted characterization factors for biodiversity impact, which is expressed as potentially PDF of species per FU

between organic and conventional milk can largely be explained by the fact that conventional cows are fed about three times more soy products than organic cows to produce the same amount of milk.

3.4 Total biodiversity damage potential

The total BDP is about 60 % lower for organic compared to conventional milk. The share of the transformation impact on the BDP amounts to about 30 and 20 % for conventional and organic milk, respectively (Fig. 6a). From all feedstock, soy showed the largest impacts on biodiversity. Although soymeal comprised only about 10 % of the conventional concentrate feed, it caused almost 40 % of the BDP. In the case of organic milk, soy constituted only 4 % in the organic concentrate feed, but was responsible for 25 % of the BDP.

To account for differences in absolute species numbers and conservation value between ecoregions, we calculated a biodiversity weighing factor. For the ecoregions which are relevant for the production of Swedish milk, the normalized weighing factor ranged between 4.3 for the Sarmatic mixed forest and 31.7 for the Cerrado (Table 3). The ratio in the weighted BDP of organic to conventional milk was slightly decreased from 43 to 36 %. The impact of rapeseed from the Baltic mixed forest, which was negligible without the weighting system, became obvious. After weighting, soy largely dominated results of the

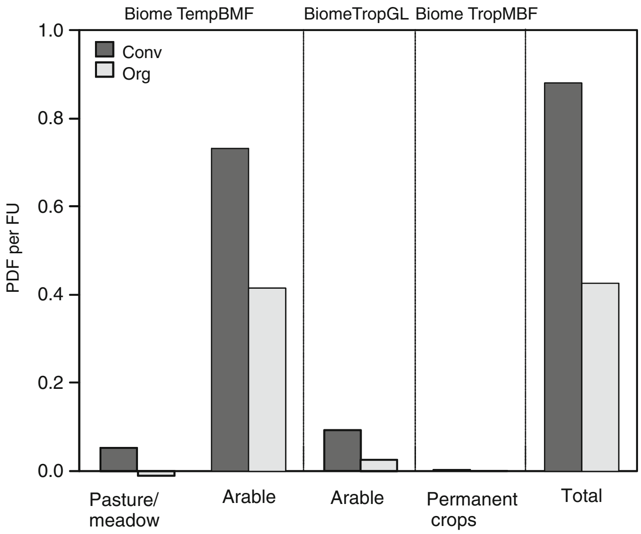

Fig. 4 Occupation impact of conventional and organic milk illustrated per land use type and biome TempBMF temperate broadleaf and mixed forests, TropGL sub-/tropical grass-/shrublands and savannahs and TropMBF sub-/tropical moist broadleaf forests, calculated with non-weighted characterization factors for biodiversity impact, expressed as PDF of species per FU

BDP. This is due to the relatively high weighting factor of the Cerrado ecoregion where most of the soy in this study was cultivated. Fifty percent of the Cerrado has been converted, whereas only 1.1 % of its area is protected, resulting in a high CRI value.

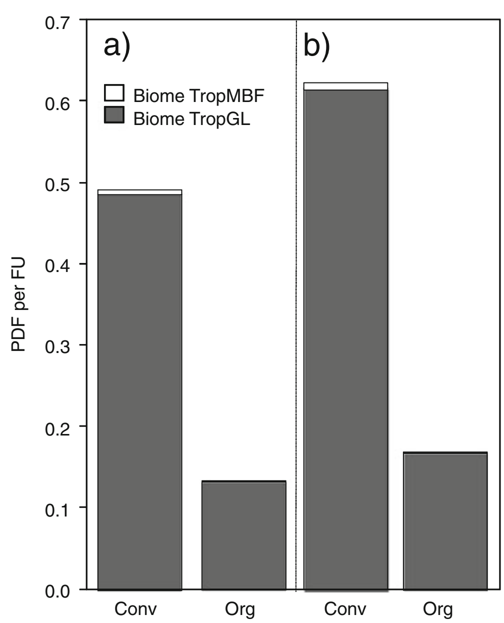

Fig. 5 *Transformation impact* calculated with regeneration times based on **a** empirical data and **b** expert estimates of biomass regeneration by Dobben et al. (1998), calculated with non-weighted characterization factors for biodiversity impact, expressed as PDF of species per FU. *TropGL* sub-/tropical grass-/shrublands and savannahs; *TropMBF* sub-/tropical moist broadleaf forests

4 Discussion

For some, organic agriculture is seen as solution for environmentally friendly food production, while for others, it is an inefficient and resource-intensive production system. To better understand the involved trade-offs, quantitative comparisons are needed. To our knowledge, this is the first study trying to quantify and compare the land use impacts on biodiversity of organic and conventional farming on a product level, including impacts of imported feedstock. While organic milk required about double the area than conventional milk, the impacts on biodiversity were less than half for organic compared to conventional milk. The impacts on biodiversity per organically farmed area were considerably smaller for conventional farming and less fodder crops originated from sub-/tropical countries where forest conversion to agricultural land uses is prevalent. These reduced impacts of organic milk could by far outweigh the larger area requirements.

Trying to express such complex issues as global land use impacts on biodiversity in one number is inherently accompanied by large natural variation and uncertainties. In the following, we want to discuss the sensitivity and robustness of results. In addition, we will discuss methodological choices made in this study, such as how to quantify biodiversity impact and the amount of transformed area, the selection of the indicator for regeneration times and the appropriateness of the weighing factor. Finally, we will reflect on the transferability of results of this study to milk produced in other countries.

4.1 Sensitivity analyses

This paper combined data from a Swedish LCA study, global literature on plant species richness and FAO statistics. All databases exhibit diverse sources of uncertainty. Conducting a full Monte Carlo uncertainty analyses was not possible in this study as information on uncertainty or variability was not available for all parameters. Therefore, to assess how the uncertainty of selected parameters influenced the results, a sensitivity analysis was performed (see Fig. 7 for an overview).

Inter-farm variation is rarely quantified in LCA studies on agricultural products, but can be considerable. Land use inventory data for this case study came from a survey of nine conventional and six organic dairy farms from the same region within Sweden. For these farms, the variability within the land occupation of both farming practices is remarkable; the farm with highest land use needs twice the area per functional unit than the one with lowest, in both farming practices, respectively. To test the robustness of the results, we compared milk from the organic farm with the largest land use and milk from the conventional farm with the smallest land use (see Fig. 7 scenario S1). In this latter case, the organic farm had a five

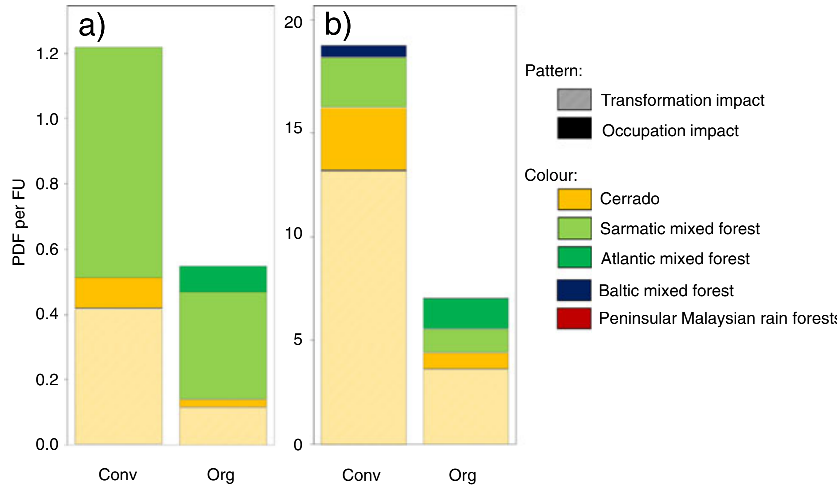

Fig. 6 *Biodiversity damage potential* (sum of occupation and transformation impact) calculated **a** without and **b** with the biodiversity weighting factor per ecoregion, expressed as PDF of species per FU. Be aware of the different scales of the axes

times larger land use than the conventional farm. This represents an extreme case for Europe, as in the published studies of other countries differences in land requirements between organic and conventional milk were less pronounced (Thomassen et al. 2008). In this case, the direct land occupation impact of organic milk from the farm with the largest land use was slightly higher than conventional milk from the farm with the smallest land use. Understanding the factors causing this inter-farm variability would be an important topic for future research as it might offer optimisation potential for both farming practices.

In agricultural LCA studies, yields are often one of the determining factors. In this current study, organic compared to

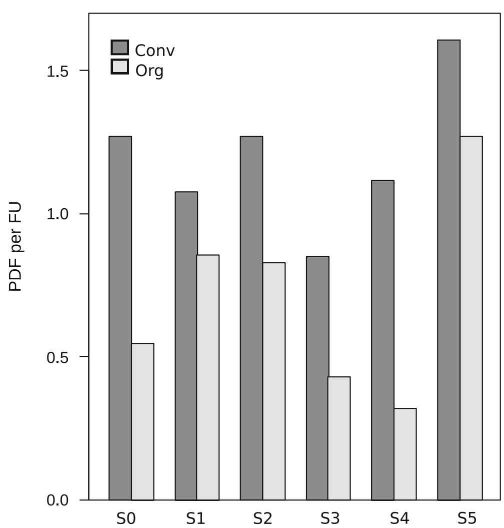

Fig. 7 *Sensitivity of biodiversity damage potential* (sum of occupation and transformation impact) per FU for scenario S0 to S5 (*S0* standard scenario used in this study; *S1* comparing organic farm with highest land use with conventional farm with lowest land use; *S2* two times lower organic yields than used here; *S3* South American soybean meal substituted with North American soybean meal; *S4* ley classified as pastures and meadows instead of arable land; *S5* using third quartiles instead of medians of characterization factors of occupation)

conventional yields were estimated to be 20 to 50 % lower per crop. Compared to yield differences ranging between 5 and 34 % found in a global meta-study (Seufert et al. 2012), our yield estimates are conservative. Even if the yields of organic crops were on average about two times lower than adopted in this study (assuming a constant composition and origin of feedstock for both alternatives), the organic milk would still be favourable in terms of impacts on biodiversity (see Fig. 7 scenario S2). Purchased feedstock is often sourced from various countries. As we did not have data for all world regions on plant species richness of organic and conventional agriculture, we could not directly test how the results would change assuming a different origin. As soy meal was responsible for about 25 % of the total BDP of organic and about 40 % of conventional milk, the total direct land use impacts could change significantly, when sourcing soy meal from another country, such as the USA, China or India. According to the applied methodology, in those countries, no transformation impact would have been allocated to soy, as in these countries the area of arable land remained stable for the last two decades (FAOSTAT 2012a) therefore decreasing the BDP (see Fig. 7 scenario 3).

Uncertainty also lies in the classification of land uses. Leys cultivated with temporary grasses and legumes grown on areas set aside were classified as arable land in this study. The reason for this decision was that leys were cultivated on average for 3 to 8 years in a crop rotation system with regular ploughing. According to Koellner et al. (2013a, b), the latter is an important differentiating factor from pastures/meadows in the land use and land cover classification used in this study. A Norwegian study found that the species richness of leys reached levels comparable to semi-natural grassland not before 30 years after last ploughing (Austrheim and Olsson 1999). However, ley grasslands might have a higher species richness than average arable fields cultivated as monocultures (Andreasen and Stryhn 2008). Thus, we performed a

sensitivity analysis classifying all the ley area as meadow/pasture. This decreased the occupation impact of conventional and organic milk about 22 and 46 %, respectively, and resulted in even more pronounced differences between organic and conventional milk (see Fig. 7 scenario S4). However, up to now almost no studies were conducted on the plant species richness of leys. Thus, more research is needed to improve reliability of biodiversity impact assessments dealing with land use on leys.

An in-depth uncertainty assessment of direct land use impacts on biodiversity of globally traded cocoa products showed that the CF_{Occ} were by far the largest contributor to variance (Mutel et al. 2013). In this study, the variance of CF_{Occ} was also considerable, even ranging from positive to negative impacts for pasture and arable land in the temperate biome and for permanent crops in the tropical biome (see Fig. 1). Part of this variation might be due to the fact that different data sources had to be used for the plant species richness of the different land use types and of the reference. Studies directly comparing the species richness of organic agriculture and a reference within the same region were not available. Consequently, data points of land use types had to be randomly combined with (semi-) natural reference data points from the same biome to generate CF_{Occ}, possibly resulting in plant species richness data from species-poor regions being combined with (semi-) natural references from species-rich regions in the same biome.

We tested the sensitivity to CF_{Occ} by recalculating the BDP results based on the third quartiles of CF_{Occ} Using the third quartiles instead of medians of CF_{Occ} increased the BDP for conventional and organic milk by about 120 and 230 %, respectively (see Fig. 7 Scenario S5). In this case, organic milk still had only half of the impact of conventional milk. When using the first quartile of CF_{Occ} for conventional land use types and the third quartile of CF_{Occ} for organic land use types, the result changes and the BDP of conventional milk would be lower than the one of organic milk. Thus, future research should aim to decrease the uncertainty of CF_{Occ} by possibly combining data points on species richness of land use types with data points on (semi-) natural references from the same region. However, a large variation will possibly still remain, as biodiversity shows a strong natural heterogeneity.

4.2 Quantification of biodiversity impacts

The strong difference in plant species richness between organic and conventional farmed land found in this study has also been found in other studies (e.g. Bengtsson et al. 2005; Fuller et al. 2005; Gomiero et al. 2011). A meta-analysis by Hole et al. (2005) on the influence of organic agriculture on biodiversity reported that differences between the two farming practices were especially pronounced for plants, birds and insects. However, for other taxonomic groups, such as earthworms or beetles, the influence of organic farming on species richness was less clear, as studies reported both positive and negative effects (Hole et al. 2005). Another meta-analysis by Bengtsson et al. (2005) found that the effect of organic agriculture was largest on studies on the plot scale, whereas on the farm scale results were still significant, however less pronounced. This might suggest that the kind of data that were chosen in this study (i.e. plant species richness on field scale or smaller) made results more clear than it could have been in the case if all organism groups had been studied. However, plants were the only taxon for which sufficient data were available for all relevant biomes and land use types. Moreover, two studies (Haas et al. 2001; Schader et al. 2010), which compared biodiversity impacts of organic and conventional milk production systems on farm scale, also found lower impacts of organic dairy farms. However, their approach did not allow the quantification of impacts related to off-farm land use which this study showed to be responsible for a considerable part of the BDP of milk.

Another factor which seems to have an influence on the species richness of agricultural land is the surrounding landscape. In their meta-analyses, Bengtsson et al. (2005) concluded that positive effects of organic farming will more likely occur in intensively used agricultural land, but not inevitably in small-scaled mosaic landscapes with a high share of natural vegetation. Thus, the results found here seem to be more representative for the highly agriculturally used landscapes found in middle Europe.

4.3 Calculation of transformed area

The quantification of area transformed for land use is largely based on FAO statistics. The database has the advantage of being easily applicable and consistent across countries (Milà i Canals et al. 2013). However, one has to be aware of several shortcomings of the database in this context. First, the lack of forest area data for the last 20 years poses a problem for studies like this one. Calculations can either be based on extrapolations of existing data or periods shorter than 20 years. This influences the results (compare with Milà i Canals et al. 2013). A second shortcoming is that the only natural terrestrial vegetation type included is "forest" (see Section 2.2 for the definition). All areas which are not classified as agricultural land or forest area are compiled in "other land" which e.g. sums up to 37 % of the country area in the case of Argentina. Hence, the assessment of conversion from more grassland-type vegetation is difficult. This study tried to overcome this problem by assuming that the entire increase of the land use type of a crop was due to direct land transformation from natural vegetation in case no other agricultural land use type decreased in area during the respective time period in the country. Furthermore, uncertainty of the FAO database is high as many data are "FAO estimates" or "unofficial figures".

Thus, in some cases it could be appropriate to compare FAO data with national inventory data, as performed for land use estimates by Ramankutty et al. (2008). Moreover, the method does not consider indirect land use change (iLUC) in other countries (Schmidt 2008b; Milà i Canals et al. 2013) which might be relevant in comparing different farming practices. It is possible that organic products have a higher iLUC as their yields are lower than conventional products (Seufert et al. 2012). However, in the scope of this study it was not possible to model worldwide socio-economic impacts of lower organic yields compared to conventional fodder crops on iLUC in other countries. Including iLUC into environmental impact assessments has only developed in the last years and was mainly limited to calculations of carbon emissions due to biofuel cultivation (e.g. Kloverpris and Mueller 2013). Methodological considerations and assumptions concerning iLUC are highly debated (Kloverpris and Mueller 2013; Hertel et al. 2010; Mathews and Tan 2009) and estimates vary considerably (Plevin et al. 2010). Nevertheless, it is expected that parameter values used to model iLUC of biofuels are not comparable to the ones that would be needed to model iLUC of organic crops. One parameter which is different is price: organic milk is about 15–83 % more expensive in the European Union (Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management 2011) which is known to decrease milk consumption (Andreyeva et al. 2010) and thus expected iLUC. However, the socio-economic cause–effect chain of converting agricultural land to organic land is complex and potential impacts on land management in other countries are barely understood. Moreover, not enough data to calculate CF_{Occ} or regeneration times for other biomes where natural ecosystems are converted for agricultural use were available. Nevertheless, including iLUC into the assessment of greenhouse gas emissions of biofuels was shown to cancel out their benefit (e.g. Hertel et al. 2010). Therefore, in future comparisons of different farming practices related to land use change should include this aspect.

4.4 Indicator for regeneration time

Up to now, biomass estimates (e.g. Milà i Canals et al. 2013; Schmidt 2008a) were used in LCIA as indicator for the regeneration of biodiversity as a whole, in contrast this study also used empirical data of plant species richness. The similarity of the results for transformation impacts (see Fig. 5) calculated with empirical data in comparison with biomass estimates by Dobben et al. (1998) is astonishing. Dobben et al. (1998)´s estimates depend only on latitude and altitude, whereas the regeneration of species richness is also modified by factors such as the natural disturbance regime (Brown and Lugo 1990), soil nutrients or the proximity to patches of natural vegetation in the surrounding landscape (Prach and Rehounkova 2006). But what takes much longer than the regeneration of biomass or vascular plant species richness are the regeneration of species composition (Aide et al. 2000; Pascarella et al. 2000; Dunn 2004), species richness of other taxa (Barlow et al. 2007; Berry et al. 2010) or endemic species richness (Liebsch et al. 2008). A recent meta-analysis of empirical studies showed that primary tropical forests are irreplaceable in terms of biodiversity (Gibson et al. 2011). A practical solution for this problem in LCA applications could be to assume a very long regeneration phase (Koellner and Scholz 2007). Koellner et al. (2013a, b) recommended to limit the modelling period in land use impact assessments to 500 years. Incorporating such long regeneration times would increase the share of transformation impacts on the overall BDP considerably. Ultimately the choice of indicator depends on the aim of biodiversity LCIA (see Michelsen 2011). Depending on whether we focus on ecosystem services, functional diversity should be protected (see also de Baan et al. 2013) which correlates well with plant species richness (Petchey et al. 2004). However, if we focus on the conservation or the potential prospective economic value (e.g. medicinal plants) of biodiversity, endemic species richness or species composition of primary habitat should be chosen. A meta-analysis on empirical regeneration times of species richness for different world regions and taxa will become available soon (Curran et al. in press) which will improve reliability of results as uncertainty of regeneration times used in this study is high due to few available data points (see Table 2). Regeneration times in this study (see Table 2) of tropical biomes are not directly comparable to the numbers found for temperate biomes as the compared land use types for the biomes differ. Regeneration of species richness on arable land is expected to take much longer than on pastures/meadows or permanent crops as the human intervention is much more intense due to regular ploughing.

4.5 Biodiversity weighting system

To account for differences in absolute species numbers as well as conservation value between ecoregions, a weighting factor was introduced, as recommended in de Baan et al. (2013). The factor proposed here is easy to apply and consistent over ecoregions on a global level. However, up to now only information for amphibians, reptiles, mammals and birds could be included. Thus, as soon as data on endemic plant species or on other important species groups such as arthropods become available, additional taxa should be included in the calculation of a weighting system to improve the information value.

Here, vulnerability was assessed as the ratio of converted to protected ecoregion area. Yet, converted areas can also be a main target of nature conservation for example the species-rich grasslands developed after deforestation and subsequent extensive grazing and mowing in Europe (Poschlod and Wallis de Vries 2002). Therefore, the vulnerability assessment

might be improved by calculating the ratio of natural and extensive land uses to protected ecoregion area. Including the areas of extensively used land might have resulted in a higher biodiversity weighting factor of the Sarmatic mixed forest. Currently, data on areas of extensive or traditional land uses are not available on ecoregion scales. Nevertheless, results of the introduced weighting scheme in this study showed that the Cerrado exhibits a considerably more valuable biodiversity than European ecoregions. Accounting for this remarkable richness is essential in the assessment of biodiversity impacts in LCAs.

4.6 Transferability to other temperate countries

Results for occupation impacts of milk found in this study cannot be transferred easily to milk produced in other countries. When comparing the land use inventories of various European milk LCA studies, the variation in allocation procedures and assumptions is posing a difficulty (De Boer 2003; De Vries and De Boer 2010; Yan et al. 2011). The sources of feedstock vary, where some studied farms source concentrate feed components from countries such as India (De Boer et al. 2013) or Australia (Thomassen et al. 2008) The biggest difference across countries seems to be their share of pastures/meadows on the total land use. As CF_{Occ} of pasture/meadow are relatively low, high proportions of this land use decrease the occupation impact. In Sweden, land use relies to only 12 % for conventional and 20 % for organic milk on permanent pastures/meadows (calculation based on data in Cederberg and Flysjoe 2004). These percentages do not include the ley areas which seem to be special for Sweden but are classified as arable due to their regular ploughing. Thus, land use for Swedish milk production is based less heavily on meadows/pastures compared to Belgium with 42 % for zero-grazing (i.e. no grazing on pastures but roughage from meadows fed) conventional milk (Meul et al. 2012), Germany (Mueller-Lindenlauf et al. 2010) with 45–89 % for organic milk or the Netherlands (Thomassen et al. 2008) with 43 % for conventional and 54 % for organic milk. The highest share of permanent pastures/meadows on the overall land use for milk seems to be found in New Zealand (Basset-Mens et al. 2009) where 95 % of the land use for high-input milk and 100 % of the land use for low-input milk (i.e. no purchased feed outside the farm) is based on permanent mixed pastures. However, it has to be investigated whether CF_{Occ} generated in this study are transferable to other European countries or regions in the biome temperate broadleaf and mixed forest (e.g. New Zealand, China or USA) or to which extent other factors such as agricultural history or sensitivity of native species to agriculture cause significantly different results.

Transformation impacts found in this study seem to be transferable to milk produced in other European countries. Most middle and northern European dairy farms source their feed largely from countries where the area of agricultural production remained stable for decades (FAOSTAT 2012a). Therefore, the share of soy meal (from South America) and palm kernel expels (from South-East Asia) in the concentrate dairy feed might be a good proxy for transformation impacts. The percentage and origin of soy and palm kernel expels in conventional concentrate feed in this case study (about 10 % of 0.32 kg concentrate feed per FU) are comparable to other European countries. A Dutch study listed 12 % Brazilian soy of 0.26 kg concentrate feed per FU (de Boer et al. 2013), a French study 13.8 % soy from Brazil (Lehuger et al. 2009) and in the regional estimate of the FAO (2010) 13 %. In contrast, percentages in organic concentrate feed (5 %) in this study seem to represent values of the upper range. In Austria (Hoertenhuber et al. 2010), no soy meal or palm kernel expels were included in organic concentrates and almost none (accounting for economic allocation) in the Netherlands (Thomassen et al. 2008). Thus, transformation impact results calculated for conventional milk seem to be representative for other European countries, whereas results for organic milk seem to demonstrate values on the upper margin. Finally, the statement that organic milk has lower transformation impacts than conventional milk seems to be valid for the studies mentioned here.

5 Conclusions

Although organic land requires about double the area than conventional milk, the direct impacts on biodiversity were less than half. This illustrates the importance of differentiating CF_{Occ} depending on the land use intensity (e.g. organic versus conventional). However, as including indirect land use change in assessments of greenhouse gas emissions of biofuels was shown to be crucial, future research on the biodiversity impact of organic products should study socio-economic effects of lower organic compared to conventional yields on indirect land use change. For an application to other studies, a more complete data set on species richness of organic and conventional farms needs to be compiled, to cover more world regions. However, for studies concerned with similar countries of origin, the presented CFs can be used to approximate impacts, considering the underlying uncertainties as discussed above. To increase validity of results, more research is required especially on the impact of land use on biodiversity compared to a (semi-) natural reference in grassland biomes, which are currently gaining in agricultural importance. Future research should assess the relevance of a more fine-scaled regionalization and develop concepts on incorporating aspects such as type of surrounding landscape or history of land use, which have a considerable influence in some regions (Bengtsson et al. 2005).

In the present study, we just compared the direct land use impacts to the endpoint Biodiversity Damage Potential (Koellner et al. 2013a, b) and did not consider other midpoint-oriented impact indicators (e.g. climate regulation, biotic production, freshwater regulation). To better understand the role of organic agriculture in an environmentally friendly solution to feed the world, other environmental as well as socio-economic impacts should be considered. As was shown in this study, further methodological developments might be needed to capture the differences between organic and conventional farming systems within LCA.

Another finding of this study is that meadows and pastures are the most biodiversity-friendly feedstock for milk production. Meadows and pastures generally will have a higher species richness than arable land, assuming stocking rates are not detrimental to biodiversity. The larger the share of pastures and meadows on the land occupation of milk, the lower the impact to biodiversity will be. As a consequence, politics should create incentives for farmers to maintain pastures and meadows, if they contain high species richness. Further improvement potential exists by sourcing soy from countries without deforestation or to replace soy with another, more biodiversity-friendly crop. Results found here also stress the importance of subsidies for organic agriculture as this type of farming practice makes an important contribution to the maintenance of species richness in the agricultural landscape.

Acknowledgments We are thankful to Anna Flysjoe (Arla Foods, Denmark) who explained the allocation procedure performed in Cederberg and Flysjoe (2004). We are also grateful to Manuel Steinbauer, University of Bayreuth for statistical advice and Christel Cederberg (Arla Foods, Denmark) and Anna Flysjoe, the Countryside Survey of the United Kingdom and the Biodiversity Monitoring Switherland (BDM) for providing data. We are grateful for helpful comments by Matthias Meier, Anna Flysjoe and three anonymous reviewers. Finally, we would like to thank Daniel Kachelriess, University of Bayreuth for revising the English.

References

Aide T, Zimmerman JK, Herrera L, Rosario M, Serrano M (1995) Forest recovery in abandoned tropical pastures in Puerto Rico. For Ecol Manag 77:77–86

Aide TM, Zimmerman JK, Pascarella JB, Rivera L, Marcano-Vega H (2000) Forest regeneration in a chronosequence of tropical abandoned pastures: implications for restoration ecology. Restor Ecol 8(4):328–338

Aide TM, Zimmerman JK, Rosario M, Marcano H (1996) Forest recovery in abandoned cattle pastures along an elevational gradient in northeastern Puerto Rico. Biotropica 28(4):537–548

Andreasen C, Stryhn H (2008) Increasing weed flora in Danish arable fields and its importance for biodiversity. Weed Res 48:1–9

Andreyeva T, Long MW, Brownell KD (2010) The impact of food prices on consumption: a systematic review of research on the price elasticity of demand for food. Gov Politi Law 100(2):216–222

Angelo C, Ripley B (2012) boot: Bootstrap R (S-Plus) Functions. R package version 1.3-6. from http://cran.r-project.org/web/packages/boot/citation.html, Accessed July 2012

Austrheim G, Olsson EG (1999) How does continuity in grassland management after ploughing affect plant community patterns? Plant Ecol 145:59–74

Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management (2011) Production and commercialization of organic milk in Europe. Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management, Department III/6, Vienna

Barlow J, Gardner TA, Araujo IS, Avila-Pires TC, Bonaldo AB, Costa JE, Esposito MC, Ferreira LV, Hawes J, Hernandez MIM, Hoogmoed MS, Leite RN, Lo-Man-Hung NF, Malcolm JR, Martins MB, Mestre LAM, Miranda-Santos R, Nunes-Gutjahr AL, Overal WL, Parry L, Peters SL, Ribeiro-Junior MA, da Silva MNF, da Silva MC, Peres CA (2007) Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. Proc Natl Acad Sci U S A 104(47):18555–18560

Basset-Mens C, Ledgard S, Boyes M (2009) Eco-efficiency of intensification scenarios for milk production in New Zealand. Ecol Econ 68(6):1615–1625

BDM (2004) Biodiversity monitoring Switzerland. Indicator Z9: species diversity in habitats. Bundesamt für Umwelt, BAFU. http://www.biodiversitymonitoring.ch. Accessed June 2012

Bengtsson J, Ahnstroem J, Weibull AC (2005) The effects of organic agriculture on biodiversity and abundance: a meta-analysis. J Appl Ecol 42(2):261–269

Berry NJ, Phillips OL, Lewis SL, Hill JK, Edwards DP, Tawatao NB, Ahmad N, Magintan D, Khen CV, Maryati M, Ong RC, Hamer KC (2010) The high value of logged tropical forests: lessons from northern Borneo. Biodivers Conserv 19(4):985–997

Brooks TM, Mittermeier R, da Fronseca G, Gerlach J, Hoffmann M, Lamoreux JF, Mittermeier C, Pilgrim J, Rodrigues A (2006) Global biodiversity conservation priorities. Science 313(5783):58–61

Brown S, Lugo AE (1990) Tropical secondary forests. J Trop Ecol 6:1–32

Carey PD, Wallis S, Chamberlain P, Cooper A, Emmett BA, Maskell LC, McCann T, Murphy J, Norton LR, Reynolds B, Scott W, Simpson I, Smart S, Ullyett J (2007) 2008 Countryside survey: UK results from 2007, NERC/Centre for Ecology and Hydrology. http://www.countrysidesurvey.org.uk. Accessed June 2012

Cederberg C, Flysjoe A (2004) Life cycle inventory of 23 dairy farms in south-western Sweden. SIK-report Nr 728. The Swedish Institute for food and biotechnology, Goeteborg

Cederberg C, Mattson B (2000) Life cycle assessment of milk production—a comparison of conventional and organic farming. J Clean Prod 8:49–60

Chappell MJ, LaValle LA (2011) Food security and biodiversity: can we have both? An agroecological analysis. Agric Hum Values 28(1):3–26

Crowder DW, Northfield TD, Strand MR, Snyder WE (2010) Organic agriculture promotes evenness and natural pest control. Nature 466(7302):109–112

Curran M, de Baan L, De Schryver AM, van Zelm R, Hellweg S, Koellner T, Sonnemann G, Huijbregts MAJ (2011) Toward meaningful end points of biodiversity in life cycle assessment. Environ Sci Technol 45(1):70–79

Curran M, Hellweg S, Beck J (in press) Is there any empirical support for biodiversity offset policy? Ecol Appl

Dalgaard R, Schmidt J, Halberg N, Christensen P, Thrane M, Pengue WA (2008) LCA of soybean meal. Int J Life Cycle Assess 13(3):240–254

de Baan L, Alkemade R, Koellner T (2013) Land use impacts on biodiversity in LCA: a global approach. Int J Life Cycle Assess 18(6):1216–1230

de Boer IJ (2003) Environmental impact assessment of conventional and organic milk production—review. Livest Prod Sci 80(1–2):69–77

de Boer IJM, Hoving IE, Vellinga TV, Ven GWJ, Leffelaar PA, Gerber PJ (2013) Assessing environmental impacts associated with freshwater consumption along the life cycle of animal products: the case of Dutch milk production in Noord-Brabant. Int J Life Cycle Assess 18:193–203

de Vries M, De Boer I (2010) Comparing environmental impacts for livestock products: a review of life cycle assessments. Livest Sci 128:1–11

Delgado CL (2003) Rising consumption of meat and milk in developing countries has created a new food revolution. J Nutr 133:3907S–3910S

Dobben HF, Schouwenberg EPAG, Nabuurs GJ, Prins AH (1998) Biodiversity and productivity parameters as a basis for evaluating land use changes in LCA. Annex 1 In: Lindeijer EW, van Kampen M, Fraanje PJ, van Dobben HF, Nabuurs GJ, Schouwenberg EPAG, Prins AH, Dankers N, Leopold MF (1998) Biodiversity and life support indicators for land use impacts in LCA. Delft: Rijkswaterstaat, Dienst Weg-en Waterbouwkunde. (Publication series raw materials 1998/07)

Dunn R (2004) Recovery of faunal communities during tropical forest regeneration. Conserv Biol 18(2):302–309

EC (2007) Council Regulation (EC) No. 834/2007 of 28 June 2007 on organic production and labeling of organic products and repealing Regulation (EEC) No 2092/91. Official Journal of the European Union L 189/1, Luxembourg

European Union (2007) Rural development in the European Union. Statistical and economic information report. Directorate-General Agriculture and Rural Development, Brussels

Ewers RM, Scharlemann JPW, Balmford A, Green RE (2009) Do increases in agricultural yield spare land for nature? Glob Chang Biol 15(7):1716–1726

FAO (2010) Greenhouse gas emissions from the dairy sector. A life cycle assessment. Food and Agriculture Organisation of the United Nations, Rome

FAOSTAT (2012a) Land Resources data 1985–2009 from http://faostat.fao.org/site/377/default.aspx#ancor. Accessed Apr 2012

FAOSTAT (2012b) Crop production data 1985–2009 from http://faostat.fao.org/site/567/default.aspx#ancor. Accessed Mar 2012

Fearnside PM (2001) Soybean cultivation as a threat to the environment in Brazil. Environ Conserv 28(1):23–38

Fearnside PM (2005) Deforestation in Brazilian Amazonia: history, rates, and consequences. Conserv Biol 19(3):680–688

FiBL (2012) Organic soy area: organic area, fully converted area and area under conversion 2010 (conversion and in-conversion not necessarily complete, both at region or country level). Research Institute of Organic Agriculture FiBL, Frick, Swiss. http://www.organic-world.net/fileadmin/documents/statistics/data-tables/crops-statistics/. Accessed Jun 2012

Flysjoe A, Cederberg C, Henriksson M, Ledgard S (2012) The interaction between milk and beef production and emissions from land use change—critical considerations in life cycle assessment and carbon footprint studies of milk. J Clean Prod 28:134–142

Foley JA, Defries R, Asner G, Barford C, Bonan G, Carpenter S, Chapin F, Coe M, Daily G, Gibbs H, Helkowski J, Tracey H, Howard E, Kucharik CMC, Patz J, Prentice C, Ramankutty N, Snyder P (2005) Global consequences of land use. Science 309(5734):570–574

Fuller R, Norton L, Feber R, Johnson P, Chamberlain D, Joys A, Mathews F, Stuart R, Townsend M, Manley W, Wolfe M, MacDonald D, Firbank L (2005) Benefits of organic farming to biodiversity vary among taxa. Biol Lett 1:431–434

Gasparri NI, Grau HR, Manghi E (2008) Carbon pools and emissions from deforestation in extra-tropical forests of northern Argentina between 1900 and 2005. Ecosystems 11(8):1247–1261

Gibson L, Lee TM, Koh LP, Brook BW, Gardner TA, Barlow J, Peres CA, Bradshaw CJA, Laurance WF, Lovejoy TE, Sodhi NS (2011) Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478(7369):378–381

Gomiero T, Pimentel D, Paoletti MG (2011) Environmental impact of different agricultural management practices: conventional vs. organic agriculture. Crit Rev Plant Sci 30(1–2):95–124

Grau H, Arturi M, Brown A, Aceñolaza P (1997) Floristic and structural patterns along a chronosequence of secondary forest succession in Argentinean subtropical montane forests. For Ecol Manag 95(2):161–171

Green RE (2005) Farming and the fate of wild nature. Science 307(5709):550–555

Guariguata MR, Chazdon RL, Denslow JS, Dupuy JM, Anderson L (1997) Structure and floristics of secondary and old-growth forest stands in lowland Costa Rica. Plant Ecol 132:107–120

Haas G, Wetterich F, Kopke U (2001) Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. Agric Ecosyst Environ 83:43–53

Hansen B, Alroe HF, Kristensen E (2001) Approaches to assess the environmental impact of organic farming with particular regard to Denmark. Agric Ecosyst Environ 83:11–26

Hertel TW, Golub AA, Jones AD, O´Hare M, Plevin RJ, Kammen DM (2010) Effects of US maize ethanol on global land use and greenhouse gas emissions: estimating market-mediated responses. Bioscience 60:223–231

Hoekstra JM, Boucher TM, Ricketts TH, Roberts C (2005) Confronting a biome crisis: global disparities of habitat loss and protection. Ecol Lett 8(1):23–29

Hoertenhuber S, Lindenthal T, Amon B, Markut T, Kirner L, Zollitsch W (2010) Greenhouse gas emissions from selected Austrian dairy production systems—model calculations considering the effects of land use change. Renew Agric Food Syst 25(04):316–329

Hole D, Perkins A, Wilson J, Alexander I, Grice P, Evans A (2005) Does organic farming benefit biodiversity? Biol Conserv 122(1):113–130

IFOAM (2012) The IFOAM norms for organic production and processing. International Federation of Organic Agriculture Movement, Bonn

Kier G, Mutke J, Dinerstein E, Ricketts TH, Kueper W, Kreft H, Barthlott W (2005) Global patterns of plant diversity and floristic knowledge. J Biogeogr 32(7):1107–1116

Kloverpris JH, Mueller S (2013) Baseline time accounting: considering global land use dynamics when estimating the climate impact of indirect land use change caused by biofuels. Int J Life Cycle Assess 18:319–330

Koellner T (2000) Species-pool effect potentials (SPEP) as a yardstick to evaluate land-use impacts on biodiversity. J Clean Prod 8:293–311

Koellner T, de Baan L, Beck T, Brandão M, Civit B, Margni M, Milà i Canals L, Saad R, de Souza DM, Mueller-Wenk R (2013a) UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. Int J Life Cycle Assess 18(6):1188–1202

Koellner T, de Baan L, Beck T, Brandão M, Civit B, Goedkoop M, Margni M, Milà i Canals L, Mueller-Wenk R, Weidema B, Wittstock B (2013b) Principles for life cycle inventories of land use on a global scale. Int J Life Cycle Assess 18(6):1203–1215

Koellner T, Scholz RW (2007) Assessment of land use impacts on the natural environment. Part 1: An analytical framework for pure land occupation and land use change. Int J Life Cycle Assess 12(1):16–23

Koellner T, Scholz RW (2008) Assessment of land use impacts on the natural environment—Part 2: Generic characterization factors for local species diversity in central Europe. Int J Life Cycle Assess 13(1):32–48

Kreuter TNO (2005) Biodiversität sächsischer Ackerflächen. Schriftenr Sächs Landesanst Landwirtsch 9(10):1–97

Lehuger S, Gabrielle B, Gagnaire N (2009) Environmental impact of the substitution of imported soybean meal with locally-produced rape-seed meal in dairy cow feed. J Clean Prod 17(6):616–624

Letcher SG, Chazdon RL (2009) Rapid recovery of biomass, species richness, and species composition in a forest chronosequence in northeastern Costa Rica. Biotropica 41(5):608–617

Liebsch D, Marques MC, Goldenberg R (2008) How long does the Atlantic Rain Forest take to recover after a disturbance? Changes in species composition and ecological features during secondary succession. Biol Conserv 141(6):1717–1725

Maeder P, Fließbach A, Dubois D, Gunst L, Fried P, Niggli U (2002) Soil fertility and biodiversity in organic farming. Science 296(5573):1694–1697

Mathews JA, Tan H (2009) Biofuels and indirect land use change effects: the debate continues. Biofuels Bioprod Bioref 147:1–13

Mattila T, Helin T, Antikainen R (2012) Land use indicators in life cycle assessment: a case study on beer production. Int J Life Cycle Assess 17(3):277–286

Mayaux P, Holmgren P, Achard F, Eva H, Stibig H, Branthomme A (2005) Tropical forest cover change in the 1990s and options for future monitoring. Philos Trans R Soc B Biol Sci 360(1454):373–384

McMorrow J, Talip MA (2001) Decline of forest area in Sabah, Malaysia: relationship to state policies, land code and land capability. Glob Environ Chang 11:217–230

Meul M, Passel S, Fremaut D, Haesaert G (2012) Higher sustainability performance of intensive grazing versus zero-grazing dairy systems. Agron Sustain Dev 32(3):629–638

Michelsen O (2008) Assessment of land use impact on biodiversity. Proposal of a new methodology exemplified with forestry operations in Norway. Int J Life Cycle Assess 13(1):22–31

Michelsen O (2011) Impacts on biodiversity from land use and land use changes—did we forget the first fundamental question? In: ISIE Conference, University of California, Berkeley, June 7–10

Milà i Canals L, Bauer C, Depestele J, Dubreuil A, Freiermuth KR, Gaillard G, Michelsen O, Mueller-Wenk R, Rydgren B (2007) Key elements in a framework for land use impact assessment within LCA. Int J Life Cycle Assess 12(1):5–15

Milà i Canals L, Rigarlsford G, Sim S (2013) Land use impact assessment of margarine. Int J Life Cycle Assess 18(6):1265–1277

Mueller-Lindenlauf M, Deittert C, Koepke U (2010) Assessment of environmental effects, animal welfare and milk quality among organic dairy farms. Livest Sci 128:140–148

Mutel C, de Baan L, Hellweg S (2013) Two-step sensitivity testing of parameterized and regionalized life cycle assessments: methodology and case study. Environ Sci Technol 47:5660–5667

Myers N, Mittermeier R, Mittermeier C, da Fronseca G, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–859

Olson DM, Dinerstein E, Wikramanayake ED, Burgess ND, Powell GV, Underwood EC, D'Amico JA, Itoua I, Strand HE, Morrison JC, Loucks CJ, Allnutt TF, Ricketts TH, Kura Y, Lamoreux JF, Wettengel WW, Hedao P, Kassem KR (2001) Terrestrial ecoregions of the worlds: a new map of life on Earth. Bioscience 51(11):933–938

Pascarella JB, Aide TM, Serrano MI, Zimmerman JK (2000) Land-use history and forest regeneration in the Cayey Mountains, Puerto Rico. Ecosystems 3:217–228

Pereira HM, Leadley PW, Proenca V, Alkemade R, Scharlemann JPW, Fernandez-Manjarres JF, Araujo MB, Balvanera P, Biggs R, Cheung WWL, Chini L, Cooper HD, Gilman EL, Guenette S, Hurtt GC, Huntington HP, Mace GM, Oberdorff T, Revenga C, Rodrigues P, Scholes RJ, Sumaila UR, Walpole M (2010) Scenarios for global biodiversity in the 21st century. Science 330(6010):1496–1501

Petchey OL, Hector A, Gaston KJ (2004) How do different measures of functional diversity perform? Ecology 85(3):847–857

Plevin RP, O´Hare M, Jones AD, Torn MS, Gibbs HK (2010) Greenhouse gas emissions from biofuels´ indirect land use change are uncertain but may be much greater than previously estimated. Environ Sci Technol 44:8015–8021

Poschlod P, WallisdeVries M (2002) The historical and socioeconomic perspective of calcareous grasslands—lessons from the distant and recent past. Biodivers Conserv 104:361–376

Prach K, Rehounkova K (2006) Vegetation succession over broad geographical scales: which factors determine the patterns? Preslia 78:469–480

R Development Core Team (2011) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna

Rahmann G (2011) Biodiversity and organic farming: what do we know? Agric For Res 3(61):189–208

Ramankutty N, Evans A, Monfreda C, Foley JA (2008) Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. Glob Biogeochem Cycles 22:1–19

Sala OE, Chapin F, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, Huber-Sanwald E, Huenneke LF, Jackson R, Kinzig A, Leemans R, Lodge D, Mooney H, Oesterheld M, Poff N, Sykes MT, Walker BH, Walker M, Wall D (2000) Global biodiversity scenarios for the year 2100. Science 287(5459):1770–1774

Schader C, Drapela T, Markut T, Hoertenhuber S, Lindenthal T, Meier M, Pfiffner L (2010) Biodiversity impact assessment of Austrian organic and conventional dairy products. In: LCA Discussion Forum: Integrating biodiversity in LCA, Lausanne, Nov 19, 2010

Schmidt J (2008a) Development of LCIA characterisation factors for land use impacts on biodiversity. J Clean Prod 16(18):1929–1942

Schmidt J (2008b) System delimitation in agricultural consequential LCA. Outline of methodology and illustrative case study of wheat in Denmark. Int J Life Cycle Assess 13:350–364

Scott AJ, Morgan JW (2012) Resilience, persistence and relationship to standing vegetation in soil seed banks of semi-arid Australian old fields. Appl Veg Sci 15(1):48–61

Seufert V, Ramankutty N, Foley JA (2012) Comparing the yields of organic and conventional agriculture. Nature 485(7397):229–232

Synthesis SOER (2010) The European environment—state and outlook 2010: synthesis. European Environment Agency, Copenhagen

StatBank Denmark (2012) Crops by crop, unit and area from http://www.statbank.dk/statbank5a/SelectVarVal/Define.asp?MainTable=AFG1&PLanguage=1&PXSId=0. Accessed Jun 2012

Stolze M, Lampkin N (2009) Policy for organic farming: rationale and concepts. Food Policy 34(3):237–244

Swift TL, Hannon SJ (2010) Critical thresholds associated with habitat loss: a review of the concepts, evidence, and applications. Biol Rev 85(1):35–53

Thomassen M, Vancalker K, Smits M, Iepema G, De Boer I (2008) Life cycle assessment of conventional and organic milk production in the Netherlands. Agric Syst 96:95–107

Tilman D (2001) Forecasting agriculturally driven global environmental change. Science 292(5515):281–284

Tscharntke T, Clough Y, Wanger TC, Jackson L, Motzke I, Perfecto I, Vandermeer J, Whitbread A (2012) Global food security, biodiversity conservation and the future of agricultural intensification. Biol Conserv 151:53–59

Van der Werf HM, Kanyarushoki C, Corson MS (2009) An operational method for the evaluation of resource use and environmental impacts of dairy farms by life cycle assessment. J Environ Manag 90(11):3643–3652

Van Elsen T (2000) Species diversity as a task for organic agriculture in Europe. Agric Ecosyst Environ 77:101–109

Weidema B, Lindeijer E (2001) Physical impacts of land use in product life cycle assessment. Final report of the EURENVIRON-

LCAGAPS sub-project on land use. Department of Manufacturing Engineering and Management, Technical University of Denmark, Lyngby

Yan M, Humphreys J, Holden NM (2011) An evaluation of life cycle assessment of European milk production. J Environ Manag 92(3):372–379

Zhang W (2008) A forecast analysis on world population and urbanization process. Environ Dev Sustain 10(6):717–730

Zimmermann JK, Aide TM, Lugo AE (2007) Old fields: dynamics and restoration of abandoned farmland. In: Chazdon RL (2008) Beyond deforestation: restoring forests and ecosystem services on degraded lands. Science 320(5882):1458–1460